Does proactive biodiversity conservation save costs?

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Abstract

Ecologists usually argue for a proactive approach to species conservation – it should start before a species is endangered so that a significant risk of extinction can be avoided. In reality, however, conservation often only starts when species populations are already in a critical state. We show that this "laissez-faire approach" is not only inferior in terms of conservation; it may also be more costly. This is somewhat surprising because the costs of maintaining populations at a level at which they are not endangered occur over a longer period. However, the costs of bringing species populations back to those levels may be so high that they outweigh the costs of the pro-active approach. We develop simple cost functions that capture the main economic and ecological parameters relevant to our argument and apply them for an assessment of the costs of common hamster (*Cricetus cricetus*) conservation in Mannheim, Germany. We find that a proactive approach would have saved between €17.2 mn and €36.4 mn compared to the laissez-faire policy that was actually employed. As laissez-faire policies are widespread our findings suggest that becoming more proactive would not only be better for conservation but also generate significant cost-savings.

Key words: conservation management, conservation costs, proactive conservation, common hamster

1. Introduction

The increasing number of endangered species (MA 2005) clearly shows that society often acts at the very last moment to halt and reverse species loss. Calls from scientists, NGOs and conservation agencies about the endangerment of a species often remain unheard until the species is close to extinction (e.g. Martín-Lopéz et al. 2009). By this time, however, the species' population is often in such a critical state that it requires a high level of support to bring it back to a viable size (Scott et al. 2010). This type of population support, including for instance captive breeding and relocation of local populations can be very costly.

Such a "laissez-faire" approach or policy where a population declines until political pressure, legal obligations or any other reason forces policy makers to implement conservation measures can be observed quite often. Consider an illustrative example from Australia. Due to industrial development in the coastal regions of south-eastern Australia, wintering habitat of the Orange-bellied Parrot (*Neophema chrysogaster)* was heavily reduced and brought the population size down to a critically low number of around 150 individuals (Menkhorst et al. 1990). At this size the population is subject to various risks including genetic inbreeding, storm casualties and predation. To counteract these risks an expensive captive breeding programme - which is still ongoing - was established in 1983 to stabilise and increase the population size (Smales et al. 2007).

From a conservation point of view, a laissez faire approach is problematic because the species population temporarily suffers from an increased extinction risk (e.g. Soulé 1990). This risk could be avoided with a proactive policy which implies that conservation activities start once indications appear that a still large population is declining towards low levels with correspondingly higher extinction risks. We show that this "laissez-faire approach" is not only inferior in terms of conservation; it may also be more costly. This is somewhat surprising because the costs of maintaining populations at a level at which they are not endangered occur over a longer period. However, our argument is that the costs of bringing species populations back to those levels may be so high that they outweigh the costs of the pro-active approach (cf. Shogren et al. 1999).

For our analysis, we develop simple cost functions that capture the main economic and ecological aspects relevant to a cost comparison between the laissez-faire and proactive approache (cf. Wätzold et al. 2006, Baumgärtner et al. 2008). These cost functions are then used in a case study to compare the hypothetical costs of proactive conservation with the real-world costs incurred under a laissez-faire policy. The case study concerns the conservation of the common hamster (*Cricetus cricetus,* protected by the EU Habitats Directive) in Mannheim, Germany. In the case study, a proactive approach could have saved altogether between €17 mn and €36 mn. As laissez-faire approaches are widespread our findings suggest that becoming more proactive would not only be better for conservation but also generate significant cost-savings.

Our research is related to cost assessments of conservation options (e.g. Wätzold and Schwerdtner 2005, Naidoo and Adamowicz 2006, Naidoo et al. 2006) and analysis of

the impact of costs in the design of conservation measures, which has become a prominent topic (e.g. Ando et al. 1998, Drechsler et al. 2006, Drechsler et al. 2007). It is also related to the literature on dynamic conservation management as we are dealing with the allocation of conservation measures over time. Most of this literature recognises that conservation management has to take into account that ecological and economic parameters change over time (Newburn et al. 2005, Pressey et al. 2007). This includes changes in land use (Costello and Polasky 2004, Strange et al. 2006), in funding available for conservation (Drechsler and Wätzold 2007), in biodiversity value of land (Meir et al. 2004, Fuller et al. 2007) as well as in knowledge about such value (McDonald-Madden et al. 2008).

2. Modelling costs of species conservation

2.1 Opportunity costs and management costs

We assume that providing habitat for a species leads to opportunity costs C (per time unit) in terms of foregone benefits of using the area for economic purposes and that these costs increase linearly with increasing amount of habitat. Opportunity costs are also time-dependent. In the distant past, they were practically zero: One can imagine a situation where it was not economically rewarding to develop areas that provided habitat for the species or where the prevalent land use generated habitat for the species (like extensive agricultural land use in former times in Central Europe, e.g. Gerowitt et al. 2003).

However, with economic development it became increasingly profitable to develop areas where the species lived or to change the type of land use necessary for species survival (such as through agricultural intensification, e.g. Persson 2010) resulting in increased opportunity costs for habitat provision. We consider that such a process takes time and that the increase in opportunity costs and the loss of habitats occurs gradually.

At some point in time this habitat loss leads to a population below a certain critical level.. From then on supplementary population management measures such as captive breeding, artificial feeding, translocation between local populations, and predator control may be needed to prevent further population decline and stimulate population recovery (e.g. Drechsler et al. 1998). The overall costs of these measures are denoted as D. The critical level may be imagined as some kind of minimum viable population size (e.g. Traill et al. 2007), but we do not wish to enter here into the debate about the meaningfulness of this concept and simply use the critical level to distinguish between two states: a state in which additional conservation measures are required for population survival and a state in which they are not.

Below we model the costs as simply as possible, allowing for convenient and transparent analysis but still capturing the essentials of our arguments. First we assume that the population size $x(t)$ of the species at time t is proportional to the amount of habitat at time t. Without loss of generality we set the proportionality factor to one and equate amount of habitat and population size. Below we use population size and amount of habitat synonymously. As argued above, for a given population size x opportunity cost C increases with time. We model this by

$$C(x(t),t) = g(t)x(t) \tag{1}$$

where the average cost per unit area, $g(t)$, increases with time. We assume that g increases linearly until some maximum value a:

$$g(t) = \min\{\alpha t, a\} \tag{2}$$

where α is some positive constant. We assume that management costs per unit time are zero for population sizes above some threshold x_0, and positive for smaller x:

$$D(x(t)) = \begin{cases} bx_0 & x(t) \le x_0 \\ 0 & x(t) > x_0 \end{cases} \tag{3}$$

where b is some positive constant. Quantity x_0 may be regarded as the critical population size so that as soon as the population size x drops below x_0 additional management (D>0) is required to support the population.

2.2 Taking into account conservation policies

Our goal is to compare different conservation policies and their associated costs. To model the laissez-faire approach, we assume that for a certain period of time the viability of the species is not an issue and no conservation measure is undertaken. This policy causes the population to decline over time until some time t_c when for some political or legal reason conservation measures have to be implemented to prevent the population from going extinct. Then, habitat loss has to be stopped and population management leads to costs $D(x)$ in addition to opportunity costs $C(x,t)$. Management costs $D(x)$ must be borne until the critical population level x_0 is reached.

We start our analysis at time t=0 when the population reaches the critical level $x(t=0)=x_0$. To model the population trajectory from time t=0 till $t=t_c$ (Fig. 1) we assume a continuous exponential decline at a rate $h \ge 0$. At time t_c the population has reached a size $x_c=x_0\exp(-ht_c)$. Opportunity costs are given by eqs. (1) and (2) and we additionally assume that average costs per unit area, $g(t)$, reach their maximum at time t_c, and remain at that level for $t>t_c$, so that $\alpha t_c=a$ and

$$g(t) = a\min\left\{\frac{t}{t_c},1\right\} \tag{4}$$

At time t_c population management costs $D(x)$ add to the opportunity costs, so with eqs. (3) and (4) total costs become

$$K(t > t_c) = \begin{cases} bx_0 + ax & x < x_0 \\ ax & x \ge x_0 \end{cases} \tag{5}$$

Even with population management and the provision of new habitat a small

population cannot expand to a large size instantaneously. Population management is assumed to increase population size at a rate $r>0$. For simplicity, we assume that r is independent of the cost D (this also reflects that regardless of the amount of population management measures, the biology of the population limits the rate (r) at which the population size can increase). The population increases until a time t_f when the population is back at the critical level x_0. From then on the cost-minimising policy is to maintain the population level at x_0.

In mathematical terms the population trajectory reads:

$$x(t)=\begin{cases} x=x_0\exp(-ht) & 0\le t\le t_c \\ x=x_0\exp(-ht_c)\exp\{r(t-t_c)\} & t_c\le t\le t_f=t_c(1+h/r) \\ x=x_0 & t\ge t_f \end{cases} \qquad (6)$$

Note that the equality $t_f=t_c(1+h/r)$ follows from the condition $x(t\ge t_f)=x_0$. Figure 1 represents a numerical example with parameters h=0.5, r=0.2 and t_c=2.

*** Figure 1 here ***

To compare the costs of proactive (h=0) and laissez-faire policies (h>0) we calculate the sum K_1 of the discounted costs from t=0 to t_c, considering the population trajectory given in eq. (6) and that during this period total cost $K(t)$ is given by opportunity cost C(x,t). The cost K_2 comprises the sum of the discounted costs from $t=t_c$ to t_f and considers both opportunity and population management costs.

Next to the various model parameters, K_1 and K_2 depend on the policy parameter h. We consider continuous time and calculate K_1 as the integral of the opportunity costs $C(x(t),t)$ weighted with the discount factor exp(-δt), with δ being the discount rate. With eqs. (1), (4) and (6) we obtain

$$K_1=\int_0^{t_c} a(t/t_c)x_0e^{-ht}e^{-\delta t}dt=\frac{ax_0t_c}{e^{\delta t_c}}\left\{\frac{e^{\delta t_c}-e^{-ht_c}}{(ht_c+\delta t_c)^2}-\frac{e^{-ht_c}}{ht_c+\delta t_c}\right\} \qquad (7)$$

For the time from $t=t_c$ until $t=t_f$ the costs are the integral over the discounted sum of habitat and population management costs $C+D$ and, with eqs. (1) and (4) – (6) amount to

$$\begin{aligned} K_2 &= \int_{t_c}^{t_c(1+h/r)}(ax+bx_0)e^{-\delta t}dt=\int_{t_c}^{t_c(1+h/r)}(ae^{-ht_c+r(t-t_c)}+b)x_0e^{-\delta t}dt \\ &= \frac{ax_0t_c}{e^{\delta t_c}}\left\{\frac{e^{-ht_c\delta/r}-e^{-ht_c}}{rt_c-\delta t_c}+\frac{b}{a}\frac{1-e^{-ht_c\delta/r}}{\delta t_c}\right\} \end{aligned} \qquad (8)$$

The total cost $K=K_1+K_2$ is a function of the model parameters x_0, a, b, δ, r, and the policy parameter or control variable h. Compared to a laissez-faire policy, a pro-active policy will lead to higher costs K_1, because higher opportunity costs have to be paid,

and to lower costs K_2, because less management costs have to be paid. The model parameters determine which policy leads to lower total cost K.

A large ratio b/a means that management costs are particularly relevant, favouring a result that a more pro-active policy leads to lower costs. A large discount rate δ weights early-occurring costs high compared to late-occurring costs, increasing the relevance of K_1 (which accrue before time t_c) relative to K_2 (which accrue after t_c) again favouring a result that a more pro-active policy leads to lower costs. A high population recovery rate r implies that the population recovery time t_f - t_c is short, decreasing the relevance of K_2 and favouring the laissez-faire policy as the lower cost approach. Which policy altogether leads to lower costs depends on the interaction of various model parameters and the answer is case-specific. The following example demonstrates how the costs of two policies, a pro-active and a laissez-faire policy, can be compared.

3. Case study

3.1. Conservation problem

Our case study concerns conservation measures for the common hamster by the city of Mannheim in the German federal state of Baden-Württemberg. The common hamster is listed as an Annex IV (strictly protected) species in the EU Habitats Directive (EG 92/43/EEC). The protection of these listed species is very strict and conservation measures include the rejection, delay and modification of land development plans as well as costly management measures (e.g. BFN 2005, Eppink and Wätzold 2009). The European Commission has invoked the Habitats Directive in court cases against several member states to force them to protect species close to extinction (EC 2006).

The main habitat of the common hamster in Western Europe is arable land and agricultural intensification has been largely responsible for the decline of the population. Once found throughout Baden-Württemberg, in 2001 hamster nests were only found on 120 ha of agricultural fields near Mannheim (SM, 2002b). The spring density of hamster burrows, an important indicator for the condition of hamster populations, was at a very low value of 0.7 burrows/ha (SM, 2002a). Until 2001, no significant measures to conserve the common hamster were taken in the study area.

Several economic development projects proposed in 2001 infringed on a sizeable share of the remaining hamster habitat around Mannheim. A discussion with the European Commission subsequently forced the city of Mannheim to take measures to ensure local persistence of the species. The implications of the enforcement of the Habitats Directive in Mannheim are well described in Eppink and Wätzold (2009) and summarised in Table 1.

Table 1 here

In Mannheim, enforcement of the Habitats Directive led to several adjustments to the proposed development projects, including alterations, a delay and partial

cancellations. At the same time, several other conservation measures were implemented, e.g. individual hamsters were relocated and a breeding programme was initiated. These measures (including restrictions on economic development) would not have been necessary had the hamster population not declined below a level where the population was critically endangered and the Habitats Directive was invoked. In the terminology of the presented model, they are population management measures which generate costs D.

In addition to the above measures, a scheme was put in place to pay farmers for employing less intensive techniques, thus making their lands suitable as hamster habitat. These less intensive techniques involve e.g. leaving long stubble when harvesting, not ploughing deeper than 25 cm, and growing perennial plants. These measures allow the hamster to survive in intensely used agricultural landscapes, but reduce yields and farmers' incomes (cf. Ulbrich and Kayser 2004). Since the payments for the measures roughly reflect the cost of maintaining hamster habitat, we consider them to be opportunity costs for habitat provision C.

Mannheim decided to maintain the population management measures and the payment scheme for a 10-year period until 2010. Thereafter, the success of these combined measures will be evaluated. Our case study has all the features of a laissez-faire approach. For a long time, no action was taken although the hamster population in the study area had already reached a critical level. Then, due to the Habitats Directive, several measures were implemented to avoid any negative impact on the population and to increase its size. Below we compare the costs of the laissez-faire approach with an estimate of the costs that would have occurred under proactive conservation.

3.2 Choice of data

To estimate the hypothetical opportunity costs, K_1 of a proactive approach we use eq. (7) and require data for t_c, a, δ, and x_0 (note that for a proactive approach h=0). We consider 1980 to be t=0 because in the early 80s the first signs appeared that the hamster population in Western Europe was in decline and would be unlikely to recover without support (Nechay, 2000; Weinhold and Kayser, 2006). In 2001, public pressure resulted in conservation action, so we set t_c=21.

Each farmer participating in the current conservation programme receives €1,200 per hectare annually. This amount is taken as a value for a. The choice of the appropriate discount rate, δ, for cost analysis in public projects is controversial (for a review on discounting, see e.g. Heal, 2007). Therefore, following Eppink and Wätzold (2009), we carry out two cost assessments for K_1 with real discount rates of 2% and 7%, respectively. A rate of 2% may be considered an example of a low discount rate and 7% an example of a high rate.

Since hamster populations were not previously monitored in Mannheim, no concrete data on the initial size of hamster habitat exist. To obtain a value for x_0, we use an estimate by MEDAD (2005) that 1,500 individuals constitute a viable local population of the common hamster. We interpret this to mean that 1,500 individuals must be alive in spring, since only these hamsters will be able to procreate. We

assume that in spring each individual corresponds to one burrow, as hamsters are solitary creatures and do not share winter burrows (Weinhold and Kayser, 2006).

Ruzic (1977) provides a ranking of hamster burrow densities: low (<0,7 nests/ha), middle (2-5 nests/ha), high (6-20 nests/ha) and very high (21-50 nests/ha). Nest densities upwards of 30 nests/ha are considered outbreaks, but even values of up to 300 nests/ha have been registered (Nechay, 2000). If we assume an average density of 13 nests/ha, this implies a value of 115 hectares for x_0.

Eppink and Wätzold (2009) estimated the population management costs D and the opportunity costs C that Mannheim incurred in the period 2001-2010 (cf Table 1). We take these data to estimate the costs of the laissez-faire approach K_2 (this implies that 2010 is chosen as t_f).

3.3 Results

Inserting the data in equations (7) and (8), we estimate the costs that would have been incurred under proactive conservation from 1980 (t_0) to 2010 (t_f). Given a policy with no habitat loss (h=0), costs K_1 amount to €1.7 mn and costs K_2 to €1.2 mn, with a total cost of €2.9 mn. These numbers provide present values for 2001 (t_c) with a discount rate of 2%. With a rate of 7%, the present values of K_1 and K_2 are €2.4 mn and €1.0 mn, respectively, resulting in a total conservation cost of €3.4 mn.

For the laissez-faire approach, K_1 is zero. According to Eppink and Wätzold (2009), the costs K_2 range between €20.6 mn and €39.5 mn (present value for 2001 with discount rates of 2% and 7%, respectively). We find that the potential cost savings of a proactive approach are significant. The lowest estimate, calculated by deducting the highest cost estimate of the proactive approach from the lowest cost estimate of the laissez-faire approach is €17.2 mn. The highest estimate amounts to a cost saving of €36.4 mn.

Two of our assumptions require some brief explanations. First, we assumed a far larger area as hamster habitat under the proactive approach than the current area of 24ha on which farmers receive compensation payments. This results in a conservative estimation of cost savings of the proactive approach as if we had chosen 24ha as x_0, costs of the proactive approach would have been even lower. Second, we assumed that the 10-year conservation programme implemented in Mannheim in 2001 will have led to a recovery of the hamster population by 2010. However, an evaluation of the programme is planned for 2010 and if the hamster population has not reached sustainable levels the programme must be extended. As this will raise the costs of the laissez-faire approach the potential costs savings of proactive conservation would again increase.

4. Summary and discussion

The starting point of our analysis was the observation that a laissez-faire approach is often adopted in species conservation – policy makers only act to prevent species from extinction once the population is in a critical state. Consequently, recovery of

the population requires a comprehensive and possibly costly set of conservation measures. Ecologists have criticised this approach as the species population temporarily suffers an increased extinction risk. To avoid this, a proactive approach may be adopted, whereby conservation starts once the population threatens to shrink below a critical level. We suggest that a laissez-faire approach may not only be inferior to a proactive policy in terms of conservation, it may also be more costly. Our argument is that the costs of recovery measures may outweigh the costs of maintaining the habitat for a minimum viable population even though these occur over a longer period.

We developed simple cost functions to capture the essential economic and ecological parameters needed to compare the costs of proactive and laissez-faire policies, and found that a low discount rate, a long recovery time for the species population and a high ratio of management costs to costs of habitat provision favour a situation in which the pro-active approach is less costly. The model was used to compare the costs of the laissez-faire approach to hamster conservation in Mannheim with the costs of a hypothetical proactive approach. We estimated that a proactive approach would have led to cost savings of between €17.2 mn and €36.4 mn. Anecdotal evidence suggests that laissez-faire policies are widespread. For instance, SZ (2007) reported 10 cases in Germany alone where – similar to hamster conservation in Mannheim – the requirements of the Habitats Directive caused substantial delays and modifications of development projects).

Why is this so given that laissez-faire policies are inferior in terms of conservation and may be even more costly? A thorough analysis of this question is beyond the scope of this paper but we wish to briefly discuss some possible answers as hypothesis for further research. A general trend is certainly that species conservation is often not given a high priority in policy making which would explain that many policy makers are willing to accept the higher extinction risks that are associated with laissez-faire policies. But why do they accept possible higher costs of a proactive approach?

One possible reason is that costs of the laissez-faire approach are not certain. If there is no sufficient political or legal pressure the species population may become extinct and no conservation costs arise. Hence, the expected costs of a laissez-faire approach (costs of the approach multiplied with the probability that they actually occur) are lower than the costs of the laissez-faire itself. Another reason may be that politicians are short-sighted or are forced by voters to behave as if they were. If politicians adopt a proactive approach voters may not re-elect them because of the immediate costs. Furthermore, politicians can blame actors such NGOs or external forces such as a court for having to take costly conservation measures associated with the laissez-faire approach.

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Tables

Table 1. Measures to protect the common hamster in Mannheim (source: Eppink and Wätzold (2009))

	Compensation payments	*Conservation management*	*Rejection, modification and delay of development projects*
Species protection programme common hamster (SPPCH)	Voluntary agreements for 24 ha of agricultural land	Mapping, monitoring, breeding programme	-
Development project			
SAP Arena	-	1 km fence	Parking lot reduced by 7 ha; multi-storey garage instead of parking lot
Sport and fair area	-	-	-
IKEA	-	1 km fence	450 ha of mostly agricultural land appointed nature reserve
Sandhofen	-	1 km fence	Building of 6.4 ha delayed by one year; possible extension rejected
Hochstätt	-	-	Residential area reduced by 10 ha
Estimated costs (€)	214,453 - 263,647	769,101 - 924,881	19,587,867 - 38,294,573

Figures

Figure 1: Typical population trajectory *x*(t) under a laissez-faire approach